Interpretation of Scale Dependent Inferences from Water Quality Data

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Abstract
A survey of 15 trout streams was conducted to evaluate spatial patterns of water quality and their relationship to biophysical processes on different scales within the driftless area of southeastern Minnesota. Results suggest that subsurface geology, surface landform and land-use patterns change significantly across this physiographic region. The Hilsenhoff Biotic Index, (%) EPT, EPT:Chironomidae, nitrate-N and specific conductance were all highly correlated with these regional biophysical characteristics. Stream discharge variance, pH, and (%) sediment on riffle sites varied signficantly with differences in watershed-level land use and morphology while alkalinity, (%) leaf substrate, (%) wood substrate, (%) shredders and the scraper:collector-filterer ratio varied with reach-level channel morphology and riparian management. Scale corrected classes of monitoring variables displayed different patterns of water quality. These results support theoretical claims that aquatic ecosystems are hierarchically structured and controlled by processes operating on multiple spatial and temporal scales. Water quality monitoring networks should be designed on a scale(s) defined by management objectives and the scale(s) upon which monitoring variables respond.

Key words: scale, water quality patterns, trout streams, driftless area, biomonitoring.

Introduction

Ecological phenomena are known to respond to processes which show hierarchical order (Koestler 1967, Allen and Starr 1982; O'Neill et al. 1986, Kolasa 1989, Wiens 1989, May 1990). Biophysical processes operating within the landscape provide a hierarchical set of constraints which define the observed characteristics and dynamics of our natural resources. Thus, processes operating at levels above those of interest constrain or limit processes at lower levels within the system (Allen and Starr 1982, O'Neill et al. 1986). These factors acting within their own holon (sensu Koestler 1967) or interacting between holons comprise the dynamic processes which define hydrologic regimes, soils and vegetation and influence physiological processes, life history characteristics and community composition and function of biota within aquatic ecosystems (Frissell et al. 1986, Cummins 1988, Delcourt and Delcourt 1988, Resh and Rosenburg 1989).

Factors controlling landscape dynamics and the structure and function of stream communities may manifest themselves at multiple spatial and temporal scales (Minshall 1988, Townsend 1989, Resh and Rosenburg 1989, Ward 1989). Geologic and climatic events exert control over landscape and watershed level characteristics on a spatial scale of 100's to 1000's of square kilometers and a temporal scale of 100,000 to 1,000,000 years. Vegetation dynamics and land management practices exert controls over processes operating over landscapes and watersheds on spatial scales of 10's to 100's of square kilometers and temporal scales of 100's to 1000's of years, while management and natural processes operating along the stream corridor determine inputs of organic matter, light energy and temperature regimes over spatial scales of 1 to 100 square meters and temporal scales of weeks to months (Frissell et al. 1986, Delcourt and Delcourt 1988, Minshall 1988).

Until recently, most efforts to define factors controlling benthic communities in streams have focused on watershed, reach and microhabitat level processes operating over temporal scales of days to months (Resh and Rosenburg 1989). In addition, these studies have focused on biological responses to processes operating on one spatial and/or temporal scale (e.g., Fisher 1987, Peckarsky 1986,1987). Despite excellent attempts to introduce hierarchy theory and the importance of scale to stream ecology (Frissell et al. 1986, Cummins 1988, Minshall 1988, Townsend 1989, Ward 1989), few attempts have been made to examine benthic data or controls over water quality monitoring variables across a number of spatial and temporal scales (except see Resh and Rosenburg 1989).

Scale phenomena also influence the design and inferences drawn from water quality investigations (Jeffers 1988). However, unlike the basic sciences, applied sciences like water quality are necessarily tied to the human perspective. This perspective (scale of human activities and influences) also operates hierarchically on multiple social and political scales and must be integrated with natural biophysical phenomenon to allow for proper monitoring and management of natural resources. In fact, it is preoccupation with the human perspective by the applied sciences which often limits the utility of monitoring data (Perry et al. 1984). Matching the scale of a natural phenomenon with the scale of a management objective is necessary to improve the efficiency and accuracy of our monitoring efforts (Schumm 1988).

The objectives of the work presented in this paper were to (1) define spatial patterns of water quality within a heterogeneous region, (2) determine the relationships between water quality monitoring variables and biophysical processes across multiple scales, and (3) compare patterns of water quality generated by variables operating on different spatial scales. We hypothesized that there would be discernible regional patterns in physical, chemical and biomonitoring variables throughout the "driftless area" (Winchell and Upham 1884). Furthermore, we hypothesized that different monitoring variables would respond at different scales of resolution (regional, watershed, reach, and riparian levels) within the driftless area, since processes controlling their dynamic behavior operate at different levels.

Study Area

Samples were collected from riffle sites on 15 randomly selected trout stream tributaries of the Root River Basin in southeastern Minnesota (Longitude 91°-93° W, Latitude 43°-44° N) (Fig. 1). This area of Minnesota was considered part of the "driftless area" by J.D. Winchell during his geological survey of the state (Winchell and Upham 1884) and falls within the "driftless area aquatic ecoregion" defined by Omernik and Gallant (1988). The climate of the study area is mid-continental with 72cm of precipitation per year, 66% occurring during the growing season (May-September). Mean air temperatures range from -10°C during the winter months to 22°C in the summer with extremes of -36°C in the winter and 36°C in the summer (Kuehnast 1972).

The Root River flows across the study area, downcutting into bedrock strata along its course to the Mississippi River (Fig. 1). Topography is largely determined by surface erosion into bedrock due to the lack of glacial till throughout much of the study area. Valley slopes exceed 35% near the Mississippi River, becoming more level along the western portion of the study area. Well drained, silty loam soils, derived from loess deposits, predominate throughout much of the study area (University of Minnesota 1973).

Natural vegetation within the region consists of maple-basswood forest in the eastern portion of the study area grading to open oak savannah in the west (Kratz and Jensen 1977). Dominant woody vegetation along stream corridors consists of willow (Salix spp.), elm (Ulmus americana L.), cottonwood (Populus deltoides Marsh) and birch (Betula spp.). Agriculture is a prominent feature of the landscape with production of corn, soybeans, alfalfa, swine and dairy cattle common in the uplands and along stream corridors (United States Department of Agriculture and Minnesota Department of Agriculture 1988).

Methods

Regional Variable

The mean elevation of spring sources above each site (ELEV) was determined from USGS topographic maps (scale 1:24000) for use as an independent variable in our analyses. ELEV served as a geology variable because changes in this characteristic imply different sources of water (i.e., geological strata). Five aquifers serve as the main sources of water for springs in the region. Streams draining the western portion of the study area originate from the karst limestone and dolomite aquifers of the Maquoketa/Dubuque and Galena formations, streams in the central portion of the study area drain from the sandstone and dolomite St. Peter and Prairie du Chien formations and streams in the eastern portion of the study area originate from sandstone aquifers of the Jordan and Franconia formations (Fig. 1). Springs were identified directly on each map or were inferred from the origin(s) of perennial stream flow on each map. ELEV data were standardized to the top of the Jordan aquifer using the information provided by Broussard et al. (1975) to correct for a westerly dip in the geological strata across the study area (Fig. 1).

Land-Use Data

Land-use information at three different spatial scales (watershed, reach, riparian) for each site was obtained from published sources and interpretation of low altitude, standard color aerial photographs. Watershed level land-use data was obtained from the Land Management Information Center (LMIC) of the Minnesota State Planning Agency (1971). The spatial resolution of this data is 40 acres (16.2 ha). Data obtained consisted of the percentage of 40 acre parcels within each section of a township which were dominated by cultivated (WCUL), pasture (WPAS) and forested (WFOR) land-use types. Aerial photographs taken during the 1987 growing season over 3 of the study watersheds were interpreted using LMIC procedures and revealed that 1969 data were satisfactory for watershed-level analyses.

Low altitude, standard color aerial photographs (1987 growing season) were obtained from county Agricultural Stabilization and Conservation Service offices to evaluate land-use at the reach and riparian levels using a modified version of the LMIC method. A twenty-five cell grid (total grid size 40 acres (16.2 ha), cell size 1.6 acres (0.65 ha)) was projected onto a color print of the section containing each site (Fig. 1). The entire grid was placed over a study site perpendicular to stream flow with the back edge of the middle cell corresponding with the location of the site. Dominant land-use in each cell was noted as was land use of the cells through which the stream flowed. Reach-level land use was estimated by calculating the percentage of cells dominated by cultivated (RECU), pasture (REPA) and forested (REFO) land-use types over the entire grid. Riparian-level land-use (RICU,RIPA, RIFO) was evaluated by calculating similar percentages for cells through which the stream flowed (Fig. 1).

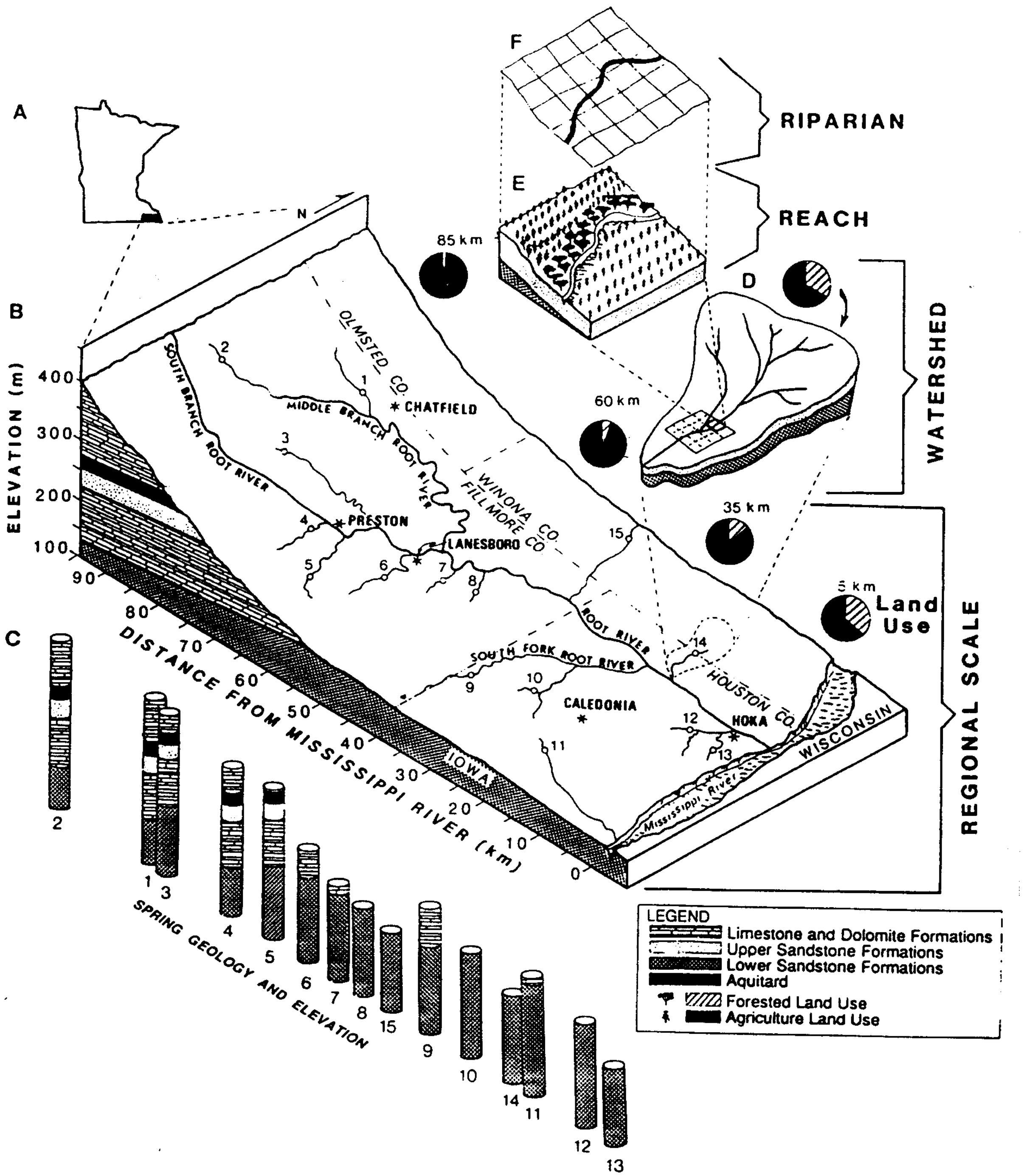

Figure 1. Diagram of study area in southeastern Minnesota showing locations of study sites, regional patterns in subsurface geology (cores) and land-use (pie-diagrams), and differences in biophysical perspective at regional, watershed, reach and riparian scales along the Root River Basin.

Table 1. Methods used in collection and analysis of water quality samples.

Variable[1]	Method	Source
NITR	Spectrophotometric	APHA[2] (1985)
ALKA	Titrimetric	APHA (1985)
PH	Electrometric	APHA (1985)
COND	YSI Model 33 S-C-T Meter	APHA (1985)
TEMP	YSI Model 33 S-C-T Meter	APHA (1985)
TURB	Hach Turbidimeter	APHA (1985)
FLCV	Sixth-Tenths-Depth Method	Platts, et al. (1983)
Substrate	% Occurrence along Transects	IN TEXT
HBIN	Duplicate, 1 min. Kicknet	Hilsenhoff (1988)
PEPT	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
EPTC	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
SHRD	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)
SCCO	Duplicate, 1 min. Kicknet	Plafkin et al. (1989)

[1]Abbreviations as defined under Methods.
[2]American Public Health Association.

Accuracy of interpretations was calculated to be 95% based on quality control procedures.

Catchment and Channel Parameters
Watershed area (AREA) and channel gradient (GRAD) from headwaters to site were determined from USGS topographic maps (scale 1:24000). Watershed gradient was estimated by calculating the absolute gradient from headwaters to site (change in elevation over channel length) using information derived from the topographic maps. Mean channel width (WIDT), depth (DEPT) and current velocity (CURR) were determined from measurements taken in the field.

Monitoring Variables
Physical and chemical water quality characteristics were evaluated on 5 randomly chosen dates in the spring and fall of 1988 (i.e., 10 repeated measures at each site). Parameters evaluated included nitrate-N (NITR), specific conductance (COND), stream temperature (TEMP), turbidity (TURB) and coefficient of variation of flow measurements (FLCV). Methods used in the determination of these parameters are shown in Table 1.

The percent occurrence of five substrate categories were determined on each riffle on the first and last sampling dates in the spring and fall (i.e., 4 repeated measures at each site). A 10 meter chain was fitted with colored flags (10cm spacing) and laid across the stream at ten equally spaced longitudinal positions along the channel. Substrate observations were made at ten locations on the chain (0.10x channel width spacing) along each of the ten transects for a total of 100 determinations per riffle. The data translated directly to percent occurrence of rock (ROCK) (diameter > 4mm), wood (WOOD), leaf (LEAF), sediment (SEDI) (diameter ≤ 4mm) and macrophyte (MACR) on each riffle.

Invertebrates were sampled at each site on the first and last sampling dates in the fall of 1988. Biomonitoring metrics evaluated from these samples included the Hilsenhoff Biotic Index (HBIN), percent of Ephemeroptera, Plecoptera and Trichoptera (PEPT) invertebrates in each sample, the ratio of EPT to Chironomidae (EPTC), percentage of shredder invertebrates (SHRD), and ratio of scrapers to collector-filterers (SCCO) (Plafkin, et al. 1989). Averages of duplicate samples collected on each date were used in further analyses (i.e., 2 repeated measures at each site). In addition, the dominant taxon in kicknet samples from each site was identified and relative abundance of these taxa were compared between sites.

Data Analysis

Abiotic gradients and biophysical relationships across the study area were identified using graphical, multiple regression and principal components analysis techniques (NH Analytical Software, 1988). Date by date Spearman Rank Correlations were calculated from repeated measures of each monitoring variable versus biophysical characteristics of each site (n=15 sites). Seasonal (n=30; 15 sites x 2 seasons) and overall means (n=15 sites) for monitoring variables were used in regression analyses to define relationships between biophysical characteristics of each site and monitoring variables at different spatial scales. If season did not contribute significantly to a regression model (t-statistic, $p>0.05$), overall site means were used in regression analyses (n=15). Model selection was based on maximizing the coefficient of determination (R^2), minimizing the residual mean square and collinearity among predictors and achieving a null residual plot through transformation of the raw data (Weisberg 1985). Results of correlation and regression analyses were used to define scale corrected groups (classes) of monitoring variables. An agglomerative cluster analysis technique was used to identify site groupings based on scale corrected classes of monitoring variables. Site means of each monitoring variable were standardized for unit differences by calculating z-scores and clustered based on squared euclidean distances between centroids of each cluster with the software package SPSS (Norusis 1988).

RESULTS

Biophysical Characteristics of Sites

ELEV above each site varied significantly (Fig. 2a) with distance from the Mississippi River. These data confirm changes in aquifer sources to trout streams distributed across the study area (Fig. 1). In addition, GRAD (Fig. 2b) and WFOR (Fig. 2c) displayed significant regional patterns across the driftless area. Lower GRAD were observed in the western portion of the study area, reflecting regional changes in topography. WFOR also decreased logarithmically with distance from the Mississippi River. These data show the increase in intensity of agricultural land-use practices on a regional scale with distance from the Mississippi River and are consistent with Minnesota state records of agricultural land-use (United States Department of Agriculture and Minnesota Department of Agriculture 1987). AREA (Fig. 2d) did not vary significantly with distance from the Mississippi River as did the other regional biophysical variables. Thus, the observed regional trends in GRAD were not merely artifacts of sampling site location within study watersheds. Regional trends exhibited by ELEV, GRAD and WFOR clearly show heterogeneity in biophysical characteristics across the driftless area.

Additional information on the biophysical characteristics of these sites is provided by the results of principal components analysis (PCA). PCA created a new set of biophysical variables through linear combinations of the original 15 variables. Eigenvector loadings of monitoring variables on each principal component (PC) indicated that there were sets of biophysical characteristics operating together on different scales (Table 2). Highest loadings on PC1 were mainly from biophysical characteristics known to vary on a regional scale (ELEV, WCUL, WFOR, GRAD). Thus, PC1 explained 31.3% of the variability in the data set and seemed to represent large scale biophysical processes.

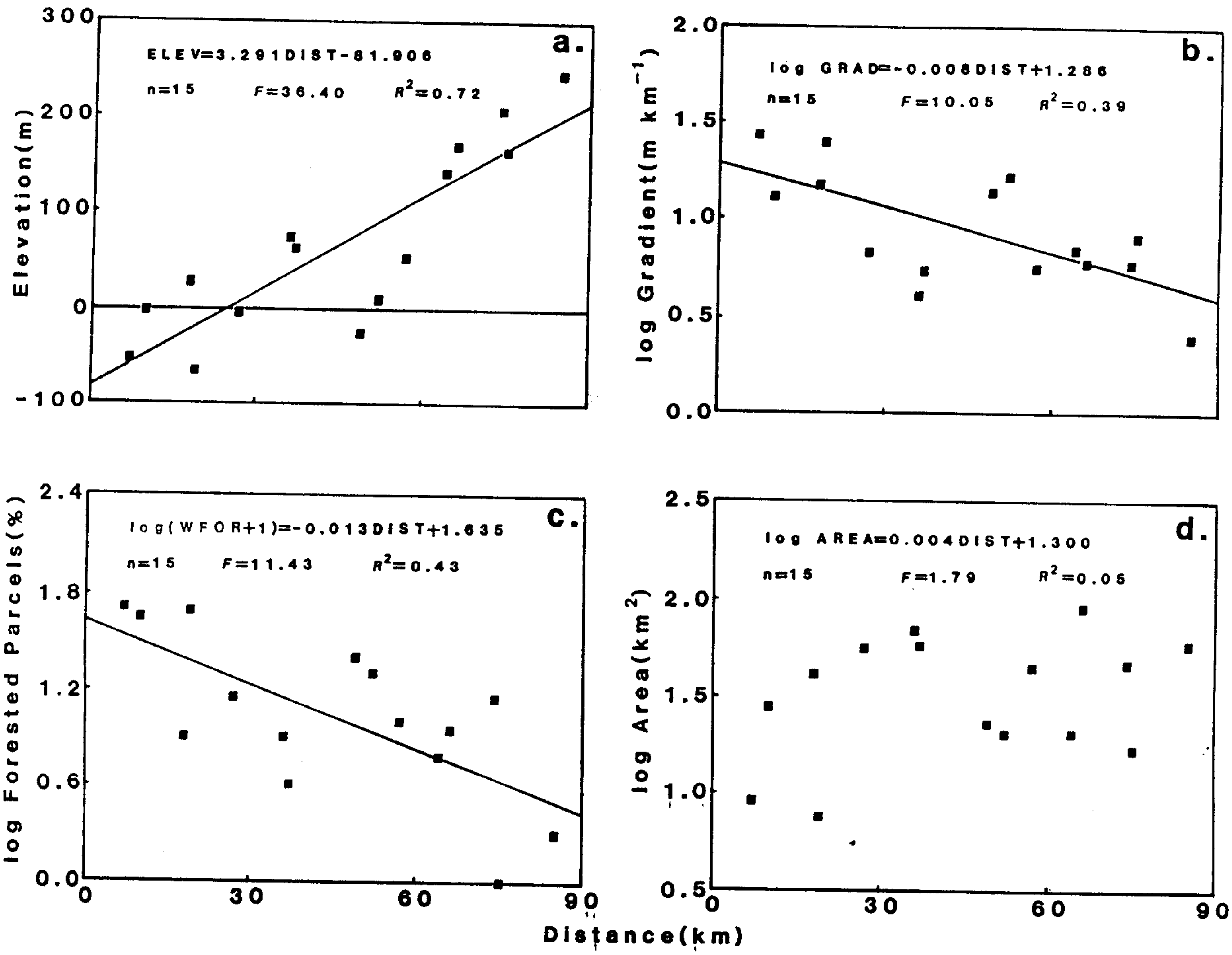

Figure 2. Regional patterns in biophysical characteristics of watersheds within the driftless area of southeastern Minnesota, (a) Relationship between mean spring elevation (ELEV) above each sampling site and aerial distance (DIST) of that sampling site from the Mississippi River; (b) Relationship between watershed gradient (GRAD) and distance from the Mississippi River; (c) Relationship between the number of 40 acre parcels dominated by forested land-use within a watershed (WFOR) and distance from the Mississippi River; (d) Relationship between watershed area (AREA) and distance from the Mississippi River.

Highest loadings on PC2 were associated with reach and riparian management practices and channel morphology (RIFO, REFO, REPA, WIDT). PC2 explained an additional 26.8% of the variability in the data set and represented local scale processes. Loadings on PC3 were highest on reach and riparian management practices (RICU, RECU, REFO). Watershed level land-use (WPAS) and morphology (GRAD) were also highly correlated with this principal component. PC3 explained 15.9% of the variability in the biophysical data set. Loadings on PC4 were only significantly correlated with channel morphology (DEPT, WIDT) at the reach level. This principal component explained 8.7% of the variability in the data set. Strong collinearity was observed among land-uses on different scales, especially

Table 2. Eigenvalues, eigenvectors and variance (% Var) explained from principal components analysis of trout stream biophysical characteristics (n=15 sites) and scales represented by each principal component.

PC	Eigenvalue	% Var	Cumulative %
1	4.689	31.3	31.3
2	4.014	26.8	58.0
3	2.382	15.9	73.9
4	1.305	8.7	82.6

VARIABLE[1]	EIGENVECTOR	SCALE
	PC1	
ELEV	-0.39	
WCUL	-0.32	
WFOR	0.32	
DEPT	-0.29	
CURR	0.32	REGIONAL/
AREA	-0.27	WATERSHED
GRAD	0.31	
REPA	-0.30	
RIPA	-0.28	
	PC2	
REPA	0.34	
RIPA	-0.31	
REFO	0.37	REACH/
RIFO	-0.39	RIPARIAN
DEPT	0.23	
WIDT	-0.35	
	PC3	
WPAS	-0.28	
RECU	-0.54	
REFO	0.37	WATERSHED/
RICU	-0.54	REACH
CURR	-0.29	
GRAD	0.24	
	PC4	
DEPT	0.42	REACH
WIDT	0.48	

[1]Abbreviations defined under Methods

reach and riparian. High collinearity among predictors within a nested hierarchy would be expected since characteristics at one scale are part of the next higher scale.

Stream Water Quality Characteristics
Most of the monitoring variables displayed large ranges in values, typical of disturbed catchments over dissolution aquifers (Table 3). NITR and TURB values occasionally exceeded water quality standards for trout waters within the state (Minnesota Pollution Control Agency 1990) and flow variance was highest in streams of the karst area within the region (Table 3).

Most of the monitoring variables were significantly correlated with biophysical characteristics at multiple scales (Table 4). NITR and COND data displayed the expected trend of low values at sites draining diffuse sandstone aquifers of the Jordan and Franconia formations and high values from the Maquoketa-Dubuque and Galena formations. TURB values showed considerable variance due to the effects of a thunderstorm which influenced samples of sites 5, 14 and 15 on one date in the spring of 1988 and men working with farm equipment in the stream at site 3 on one date during fall sampling. Thus, COND seemed to respond on a regional scale with changes in subsurface geology and regional land-use. NITR and PH seemed to be influenced by watershed-level land-use characteristics while ALKA, TURB and TEMP were highly correlated with reach and/or riparian level management practices (Table 4). FLCV was not significantly correlated with any of the biophysical characteristics.

All five substrate types on riffles displayed large ranges in values (Table 3). ROCK was the dominant substrate at riffle sites across the study area followed by MACR (Table 3). WOOD was the least common substrate type and was highly correlated with the amount of reach and riparian forest above and adjacent to a site (Table 4). Similar observations were made of LEAF material as a substrate, except the occurrence of LEAF material was strongly seasonal (see below). SEDI was prevalent as a substrate type

Table 3. Overall summary statistics for monitoring variables evaluated in Southeast Minnesota Streams (n-number of repeated measures x 15 sites).

Variable[1]	n	Mean	Median	s.e.	Range
		Physical/Chemical			
NITR (mg L^{-1})	150	3.9	3.4	0.2	0.5-10.9
ALKA (mg L^{-1})	150	266	262	2	226-333
pH	150	8.01	8.02	0.03	7.05-8.74
COND (uS cm^{-1})	150	397	386	5	275-660
TEMP (°C)	150	11.7	11.0	0.4	4.2-25.0
TURB (NTU)	150	5.12	1.80	1.37	0.30-146.00
FLCV (%)	30	19.2	15.3	2.5	2.6-66.3
		Substrate			
ROCK (%)	60	51.8	56.0	3.2	0.0-89.0
WOOD (%)	60	2.5	1.0	0.5	0.0-13.0
LEAF (%)	60	8.2	1.0	1.5	0.0-48.0
SEDI (%)	60	8.4	4.0	1.5	0.0-56.0
MACR (%)	60	28.8	22.0	3.5	0.0-94.0
		Invertebrates			
HBIN	30	3.7	3.3	0.2	1.8-6.3
PEPT (%)	30	55.2	57.3	4.5	1.5-93.9
EPTC	30	9.1	5.8	2.2	0.02-63.7
SHRD (%)	30	12.1	7.2	2.8	0.0-67.3
SCOO	30	0.8	0.3	0.2	0.0-4.6

[1]Abbreviations as defined under Methods.

at many of the agricultural sites and was most highly correlated with riparian land management (Table 4). ROCK and MACR were most highly correlated with watershed and reach-level management practices. ROCK was more abundant at agricultural sites with high current velocities while MACR tended to be more abundant at forested sites with lower current velocity. Ranunculus aquatilis (Chaix), Veronica connata var. glaberrima (Pennell) and Nasturtium officinale (R. Br.) were the macrophyte species occurring most frequently at all sites across the study area.

HBIN values ranged from "fair" (6.25) to "excellent" (1.77) indicating that some of these trout streams were influenced by significant organic loading. The PEPT in kicknet samples ranged from 1.5 to 93.9% and EPTC in kicknet samples ranged from 0.02 to 63.67. These three metrics suggested differences in invertebrate community structure between sites and all three were highly correlated with regional changes in subsurface geology and watershed morphology (Table 4). Regional changes in benthic community structure were confirmed by examining the dominant invertebrate taxa at each site. Two western sites (1,5) were dominated by the chironomids Tanytarsus sp. (35%) and Rheotanytarsus sp. (54%), site 3 was dominated by the mayfly Baetis tricaudatus vagans (McDunnough) (42%), and site 4 was dominated by the caddisfly Cheumatopsyche spp. (17%). Empirically derived tolerance values to organic pollution for these taxa were 6, 6, 2 and 5 on a scale of 0-10 (Hilsenhoff 1987). The invertebrate communities of three centrally located sites (6, 8, 10) and one western site (2) were dominated by Optioservus fastiditus (LeConte). This

Table 4. Highest Spearman Rank correlation[1] between each monitoring variable and the most frequently correlated predictor at each scale based on date by date correlations of monitoring variables with biophysical characteristics.

Variable[2]	Regional[2]	Watershed[2]	Reach[2]	Riparian[2]
NITR	ELEV(0.79)	WFOR(-0.91)	CURR(0.77)	RIFOR(-0.64)
pH	ELEV(-0.56)	WPAS(0.83)	REPAS(-0.46)	RIFOR(0.39)
TURB	ELEV(0.50)	WPAS(0.58)	RECUL(0.74)	RIPAS(0.62)
ALKAL	ELEV(0.55)	GRAD(0.50)	RECUL(-0.69)	RICUL(-0.66)
COND	ELEV(0.81)	WFOR(-0.73)	REPAS(0.68)	RIPAS(0.67)
TEMP	ELEV(-0.67)	GRAD(0.58)	REFOR(-0.68)	RICUL(0.56)
FLCV	NO SIGNIFICANT CORRELATIONS WITH SITE VARIABLES			
ROCK	-	WPAS(0.71)	CURR(0.48)	RICUL(0.60)
WOOD	-	WFOR(0.52)	REPAS(-0.53)	RIFOR(0.52)
LEAF	ELEV(0.52)	GRAD(-0.49)	REPAS(-0.76)	RIFOR(0.41)
SEDI	ELEV(-0.38)	AREA(-0.60)	REFOR(-0.54)	RIFOR(0.62)
MACR	-	WPAS(-0.63)	RECUL(-0.69)	RICUL(-0.64)
HBIN	ELEV(0.62)	GRAD(-0.69)	CHNWD(0.55)	-
PEPT	ELEV(-0.64)	GRAD(0.53)	CHNWD(-0.55)	-
EPTC	ELEV(-0.56)	AREA(-0.68)	CURR(0.58)	-
SHRD	ELEV(0.50)	WFOR(-0.38)	RECUL(-0.74)	RICUL(-0.42)
SCOO	-	AREA(0.42)	REFOR(0.57)	RICUL(-0.61)

[1]Correlations presented in table statistically significant ($p<0.10$) based on quantiles for the Spearman's Test Statistic (Conover 1980).
[2]Abbreviations as defined under Methods.

elmid beetle contributed 19-27% of the cumulative number of invertebrates sampled at each of the sites and has a tolerance value of 4 on a scale of 0-10. Site 7 kicknet samples were dominated by the caddisfly Micrasema kluane (Ross and Morse) (50%) and the invertebrate communities of the eastern sites within the study area (sites 11-15) were all dominated by Brachycentrus occidentalis (Banks). This caddisfly contributed 21-55% of the cumulative abundance of all taxa collected on both dates at each of the eastern sites. Both of these brachycentrid caddisflies have tolerance values of 1 on a scale of 0-10. Thus, invertebrate communities in the western portion of the study area were dominated by taxa which were moderately tolerant to organic enrichment (except sites 2,3) while communities of eastern sites were dominated by taxa which exhibited low tolerance to high organic loadings.

While community structure seemed to be influenced primarily at the regional and watershed levels, invertebrate community function was more highly correlated with local reach/riparian level processes (Table 4). SHRD and SCOO were positively correlated with the extent of forest development at the reach and riparian-levels and negatively correlated with agricultural land-uses on these same scales.

Regression Relationships

Statistically significant relationships were observed for all monitoring variables with one or two biophysical characteristics (Table 5). However, the variance explained by these models was quite variable (range of R^2= 0.17 to 0.92). Six of the 17 monitoring variables displayed significant seasonal differences (i.e., PH, TURB, COND, TEMP, FLCV and LEAF). Of these six monitoring variables, only LEAF displayed higher values in fall samples. The remaining monitoring variables had significant amounts of variability explained by a combination of watershed, reach and riparian level biophysical site characteristics (Table 5). More than half of the regression equations explained 40% or more of the variance in the monitoring data.

Our monitoring variables can be divided into groups based on the predictors in each model (Watershed, Watershed/Reach, Reach/Riparian). Thus, 7 of the monitoring variables had significant amounts of their variance explained by biophysical characteristics on the watershed level (NITR, PH, FLCV, SEDI, HBIN, PEPT, EPTC), 2 monitoring variables by a combination of characteristics on the watershed and reach levels (COND, TEMP) and 8 variables by characteristics on the reach and riparian levels (TURB, ALKA, ROCK, WOOD, LEAF, MACR, SHRD, SCOO). Combining the results of correlation and regression analyses, we delineated scale corrected classes of monitoring variables (Table 6). Four classes (regional, watershed, reach, riparian) were defined.

Patterns at Different Scales

Scale corrected classes of monitoring variables (Table 6) were used to generate site groupings. Cluster dendograms were generated using monitoring variables which had the greatest amount of their variance explained by regional, watershed, reach and riparian-level biophysical characteristics. Dendograms produced by these analyses (Fig. 3) were compared to identify and interpret differences in spatial patterns of water quality within the study area.

Each dendogram portrays a different pattern of site groupings based on the relationship between monitoring variables used to generate the dendogram and biophysical characteristics of each site. Regional patterns (Fig. 3a) generated from NITR, COND, HBIN, PEPT and EPTC monitoring data show 5 distinct site groupings. An examination of site characteristics with respect to those monitoring variables allowed interpretation of the observed pattern. Cluster 1 (sites 8, 11, 12, 13) had low NITR and COND compared to the regional median. Cluster 2 (sites 2, 6, 15) had high NITR and COND values. Cluster 3 (sites 1, 4, 9, 10) had high HBIN, low PEPT and EPTC values compared to regional medians. In contrast, the cluster formed from the combination of clusters 1 and 2 (above) had low HBIN, PEPT and high EPTC values. The cluster formed from the agglomeration of sites 3 and 5 differed from the other monitoring sites due to low PEPT values. These two sites were located in the karst portion of the study area below heavily developed watersheds.

The watershed-level dendogram (Fig. 3b) delineates two major groups of sites; group 1 (Sites 2,4,6,8,9, 10,11,13,15) and group 2 (Sites 1,3,5, 7,12,14). These two groups can be distinguished by differences in SEDI and MACR substrate at riffle sites. Group 1 sites had lower SEDI and higher MACR than group 2 sites when compared to regional median values.

Reach and riparian-level analyses produced very similar site patterns (Fig. 3c,d). Two groups of sites are easily delineated from the dendograms produced by these analyses. All sites except 3,5,7 at the reach-level and 5, 7 at the riparian level belong to one large group of relatively similar sites. These outlier sites displayed poor water quality characteristics (i.e., higher NITR, COND, ALKA and SHRD and lower WOOD) as compared to regional median values for each monitoring variable.

Thus, cluster analysis on scale corrected monitoring data provided distinctly different site groupings which could be interpreted from monitoring variables operating on different scales.

Table 5. Regression models for monitoring variables and biophysical factors in Southeast Minnesota streams (n-number of observations; R^2-coefficient of determination; F(p)- F-statistic and probability value for regression; RMS-residual mean square for regression; Season- t-statistic and significance for season effect in regression (*-p<0.05, NS-p>0.05)).

Variable[1,2]	Predictor[1,2]	n[3]	R^2	F(p)	RMS	Season
NITR	-log(WFOR+1)	15	0.84	76.2 (<0.001)	1.039	0.62,NS
pH	-WCUL -GRAD	30	0.42	7.9 (<0.001)	0.042	-3.21,*
log TURB	RECU RIPA	15	0.37	5.0 (<0.016)	0.037	-1.52,NS
log ALKA	-RECU	15	0.17	3.9 (0.047)	0.001	0.29,NS
log COND	-WFOR REPA	30	0.70	23.0 (<0.001)	0.001	-5.27,*
log TEMP	log(WPAS+1) -log(REFO+1)	30	0.81	41.7 (<0.001)	0.003	-10.43,*
log FLCV	WCUL AREA	30	0.37	6.6 (<0.001)	0.051	-3.90,*
log Rock	CURR	15	0.25	5.7 (0.017)	0.044	-0.76,NS
arcs WOOD	RECU RIFO	15	0.52	8.7 (0.002)	0.004	-1.58,NS
log(LEAF+1)	log(RIFO+1)	29	0.92	172.9 (<0.001)	0.026	18.27,*
log(SEDI+1)	-log AREA	15	0.21	4.7 (0.029)	0.164	-0.60,NS
arcs MACR	-RECU	15	0.32	7.5 (0.007)	0.063	-1.20,NS
log HBIN	-log GRAD	15	0.42	11.2 (0.002)	0.011	-
arcs PEPT	GRAD	15	0.28	6.5 (0.011)	0.054	-
log(EPTC+1)	-AREA	14	0.53	15.4 (<0.001)	0.065	-
log(SHRD+1)	-RECU	15	0.38	9.55 (0.003)	0.086	-
log(SCOO+1)	REFO	15	0.28	6.4 (0.012)	0.034	-

[1]Monitoring and biophysical variable abbreviations as defined under Methods.
[2]Data transformations included arcs=$(\sin^{-1})^{1/2}$ and log=common logarithm.
[3]Seasonal means used in regression (n=30) or overall means (n=15).

Discussion

Data presented in this paper suggest that commonly used water quality monitoring variables respond to processes operating on several spatial scales within the driftless area. Regional patterns in NITR, COND, and invertebrate community structure were highly correlated with regional trends in subsurface geology and land-use patterns. Streams in the western portion of the study area drain karst limestone and dolomite aquifers (Winchell and Upham 1884, Broussard et

Table 6. Scale corrected classes[1] of monitoring variables based on correlation and regression analyses.

Variable[2]	REGIONAL	WATERSHED	REACH	RIPARIAN
LEAF			....	_____________
WOOD		....	_____________	_____________
TURB		....	_____________	_____________
SHRD			_____________	....
SCOO			_____________	....
ALKA			_____________	
MACR		....	_____________	....
ROCK		....	_____________	....
TEMP		_____________	_____________	....
COND	....	_____________	_____________	....
NITR	....	_____________	....	....
PH		_____________		
FLCV		_____________		
EPTC	....	_____________		
SEDI	....	_____________		
HBIN	....	_____________		
PEPT	....	_____________		

[1]Class Membership Based on Regression Results- _____________
Class Membership Based on Significant Correlations- ....
[2]Abbreviations as defined under Methods.

al. 1975, Ojakangas and Matsch 1982, Singer et. al. 1983). The combination of intensive agricultural land-use within these watersheds and karst subsurface geology promotes water quality problems (LeGrand 1973, Singer et al. 1982, St. Ores et al. 1982, Hallberg et al. 1985, Wall et al. 1989) which simplify the structure of invertebrate communities by reducing or eliminating intolerant taxa (Perry et al. 1988, Troelstrup and Perry 1989, Bartodziej and Perry MS, Wilton and Perry unpubl. data). In contrast, streams in the eastern portion of the study area originate from sandstone aquifers and drain watersheds with more woody vegetation. Despite steeper gradients, these streams have the lowest NITR, COND and HBIN values and the highest PEPT and EPTC values.

These regional biophysical characteristics serve as a template over which finer grain reach and riparian processes operate (Frissell et al. 1986, Minshall 1988, Townsend 1989, Ward 1989). Local responses of NITR, COND and PH were probably related to subsurface dynamics within the riparian zone. Agricultural land use adjacent to a stream is known to reduce denitrification and plant uptake of nitrogen and promote nitrification and nutrient export to the stream channel (Vitousek and Melillo 1979, Peterjohn and Correll 1984, Pinay and Decamps 1988). Higher NITR, COND and PH values would be expected adjacent to agricultural conditions where redox potentials and movement of soluble ions are high (Green and Kauffman 1989).

TURB, ALKA and TEMP were also observed to vary with reach and riparian management in this study. These parameters are influenced directly by loss of riparian vegetation and subsequent bed and bank erosion in and adjacent to the stream channel. TURB levels increase dramatically in response to livestock grazing and cultivation adjacent to the stream

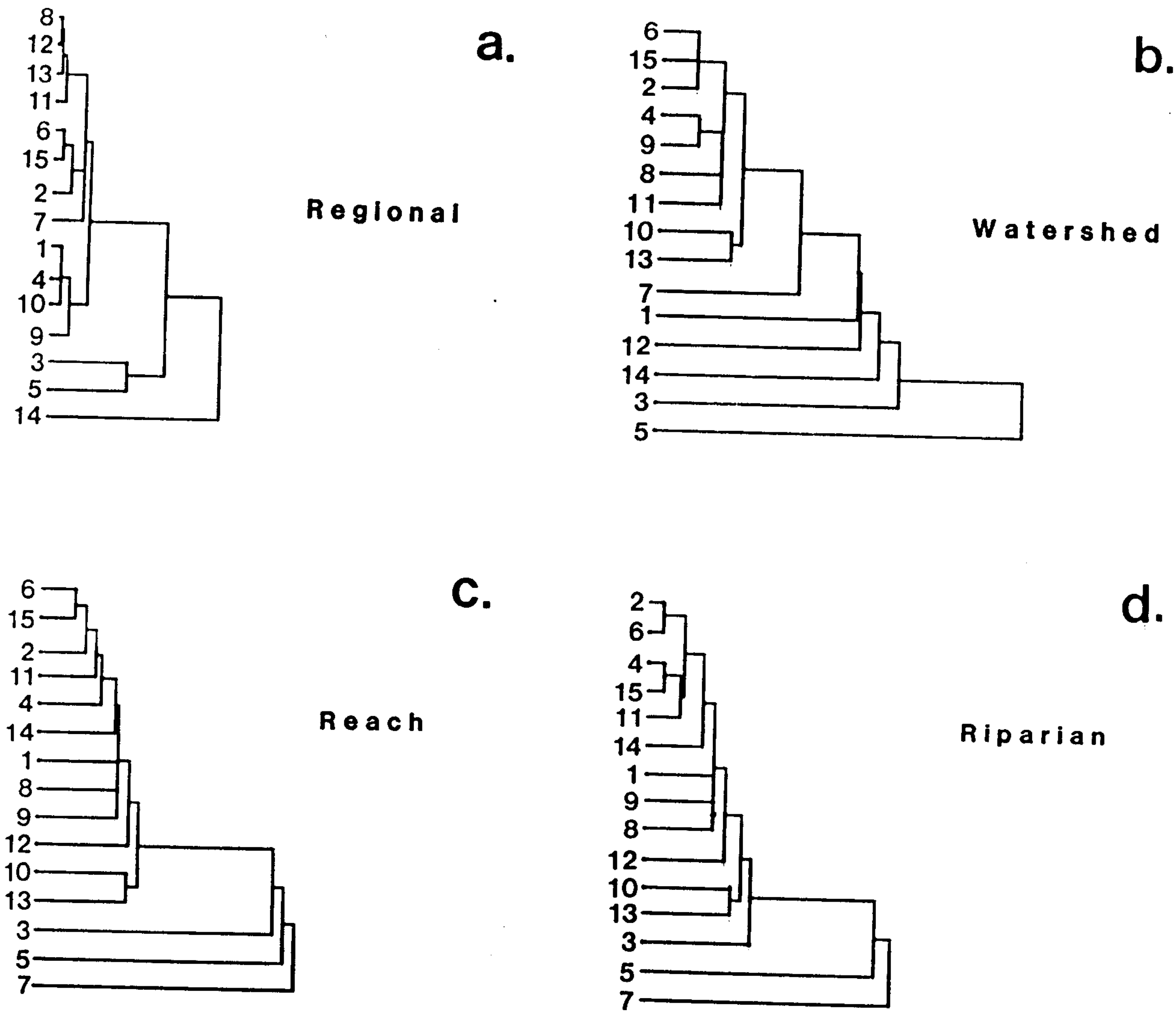

Figure 3. Results of clustering sites on scale corrected classes (see Table 6) of monitoring variables, (a) dendogram generated from regional class, (b) dendogram generated from watershed class, (c) dendogram generated from reach class, (d) dendogram generated from riparian class (Numbers adjacent to each dendogram refer to site locations as defined in Figure 1).

bank (Woodall and Wallace 1972, Karr and Schlosser 1978, Bratton et al. 1980, Menzel et al. 1984). These activities in close proximity to the stream may also increase channel width and reduce channel depth due to sedimentation of sloughed material from the stream bank (Clifton 1989). Numerous studies have noted increases in mean temperature and greater ranges in temperature regimes adjacent to agricultural areas (Karr and Schlosser 1978, Bratton et al. 1980, Dance and Hynes 1980, Menzel et al. 1984, Smart et al. 1985).

Substrate characteristics in driftless area streams are probably controlled by a combination of hydrologic processes operating on the watershed level and light and mesoscale hydrodynamics on the reach and riparian levels. ROCK and MACR substrate types were negatively correlated with one another and highly correlated with current regimes and reach and riparian management. Occurence of SEDI was negatively correlated with watershed area and positively correlated with reach and riparian management. The flashy nature of karst streams (LeGrand 1973, Hallberg et al. 1985) and prevalence of sedimentation in agricultural reaches (Karr and Schlosser 1978, Dance and Hynes 1980, Lenat 1984) would seem to explain observed patterns of these substrate types throughout the study area. WOOD material in the channel and LEAF material on the stream bottom were negatively correlated with reach pasture land and positively correlated with riparian forested land. Seasonal patterns in LEAF abundance associated with autumn abscission were also observed, suggesting that spatial and temporal patterns of availability govern the dynamics of these substrate types. Agricultural streams flowing through open riparian canopies have been shown to harbor extensive algal communities (Menzel et al. 1984, Smart et al. 1985, Bachmann et al. 1988). Thus, management of the riparian zone may directly influence functional characteristics of a stream by altering detrital inputs and primary production on a local scale within a watershed (Hynes 1975, Swanson et al. 1982).

Biomonitoring has been promoted as a useful tool in evaluating support of designated uses in water quality investigations (Lenat et al. 1980, Hilsenhoff 1982, Lenat 1988, Plafkin et al. 1989). Corkum and Ciborowski (1988) and Corkum (1989) were successful in delineating broad scale invertebrate community patterns associated with biophysical characteristics in northwestern North America. Invertebrate community structure proved sensitive to regional patterns in geology, surface land form and land-use in the driftless area. We observed low PEPT and EPTC and high HBIN values from streams draining dissolution aquifers through agricultural watersheds. Others have observed similar large scale patterns in community structure using these metrics (Welch et al. 1977, Bratton et al. 1980, Dance and Hynes 1980, Hilsenhoff 1982, Lenat 1984, Menzel et al. 1984, Hite and Bertrand 1989).

We observed shifts in the relative abundance of different feeding guilds within the invertebrate communities at our sites. Higher SHRD and SCOO values were observed adjacent to forested riparian zones. If food were a limiting resource to these insects, SHRD abundances would be expected to track the availability of LEAF material (Hynes 1975, Swanson et al. 1982, Cummins et al. 1989) while SCOO abundances would track the availability of benthic algae in the stream (Dance and Hynes 1980, Karr and Dudley 1981, Menzel et al. 1984). Ross (1963) provided evidence of regional patterns in the distribution of caddisflies (Trichoptera) related to the predominant terrestrial vegetation. Within his large scale framework, our data suggest that functional characteristics of invertebrate communities may be tightly tied to local biophysical characteristics which influence inputs and types of organic material to the stream (Hynes 1975, Swanson et al. 1982, Cummins et al. 1989).

The results of this study provide evidence of significant variability in biophysical characteristics across the driftless area in southeastern Minnesota. Subsurface geology, land surface form and land-use all vary significantly with distance from the Mississippi River. These trends in biophysical characteristics explain a significant amount of the variability in physical (COND), chemical (NITR) and biological (BIND, PEPT, EPTC) water quality monitoring variables. Regional patterns, together with local variance in reach and riparian characteristics, provide a mosaic of biophysical factors which vary at multiple spatial and temporal scales. This presents a tremendous challenge to the "aquatic ecoregion" concept which has been tested widely (Hawkes et al. 1986, Rohm et al. 1987, Hughes et al. 1987, Larsen et al. 1988, Whittier et al. 1988, Lyons et al. 1989) and implemented by state water quality agencies (e.g., Hieskary et al. 1987).

Aquatic ecoregions were originally defined to address national and large scale regional water quality issues (Omernik 1987). However, several investigators have noted problems in implementing this approach to water quality monitoring and management. Omernik and Griffith (1986) observed differences in alkalinity between seepage and drainage lakes within the same ecoregion. Hawkes (1986) found that fish ecoregions in Kansas showed little similarity to the aquatic ecoregions of Hughes and Omernik (1981). Lyons (1989) found that local habitat characteristics associated with reach and riparian management were better predictors of fish community characteristics than membership within an aquatic ecoregion in Wisconsin. Whittier et al. (1988) found poor separation of ecoregions in Oregon using periphyton and invertebrate data from streams across the state. In particular, large within-region variability occurred when valley and mountain streams occupied the same ecoregion. This challenge has been met by generalizing the "aquatic ecoregion" concept to a "regionalization" concept (Gallant et al. 1989). Under this approach, regions may be defined at any scale. Thus, heterogeneous regions may be broken up into smaller more homogeneous units. This approach has great promise if state water quality agencies can secure funding to increase the number of monitoring stations necessary for such stratification.

Properly designed water quality monitoring programs operate from well defined objectives, utilize monitoring variables which relate directly to those objectives, provide spatial and temporal information necessary to address those objectives and optimize resources to account for natural variability in measured parameters (Schaeffer et al. 1985, Perry et al. 1985). Consideration of which variables to measure is an important step in the planning process of these efforts. Many state and federal water quality agencies evaluate a list of variables at monitoring sites distributed over a sociopolitical area (state or county) and sample on a regular temporal frequency (e.g., 1 month) (Perry et al., 1984). Our data suggest that different monitoring variables are controlled by processes operating on different scales (regional, watershed, reach, riparian) and that different sets of variables may be more effective for detecting water quality problems at different scales. The variable/scale associations identified in this study may not be appropriate for other physiographic regions. In addition, this study focused on spatial patterns of water quality. Temporal dynamics in biophysical characteristics are equally important when designing a monitoring program (Wiens 1989). However, the methods used to derive scale relationships in this study could be used in other regions. Thus, scale corrected groups of monitoring variables could be identified which would (1) provide more sensitivity to a problem on a particular scale and (2) provide greater security against conflicting results due to scale incompatibility with management objectives.

Current ecological theory suggests that natural systems are hierarchically structured. The characteristics of any level of a natural system are thought to be controlled by rate processes operating on higher spatial and temporal scales (Koestler 1967, Allen and Starr 1982, O'Niell et al. 1986, O'Neill 1988). Examples of controlling processes operating within the landscape include faulting, volcanism, climatic changes, pedogenesis, weathering and erosion. These processes result in the development of controlling factors in the landscape which influence the characteristics of water resources and the biota inhabiting aquatic ecosystems. Clearly, interpretation of structural and functional patterns in nature is scale dependent and our ability to monitor and manage those resources depends on our understanding of that hierarchical structure and its dynamics.

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Literature Cited

Allen, T.F.H. and T.B. Starr. 1982. Hierarchy. Perspectives for ecological complexity. University of Chicago Press, Chicago.

American Public Health Association. 1985. Standard methods for the examination of water and wastewater. 16th edition. American Public Health Association, Washington, D.C.

Bachmann, R.W., K.J. Kortge, T.E. Robertson. 1988. Primary production in small agricultural stream. International Association of Theoretical and Applied Limnology 23: 1179-1182.

Bartodziej, W. and J.A. Perry. MS. Litter processing in diffuse and conduit springs. Hydrogiologia (IN PRESS).

Bratton, S.P., R.C. Mathews, Jr., P.S. White. 1980. Agricultural area impacts within a natural area: Cades Cove, a case history. Environmental Management 4: 433-448.

Broussard, W.L., D.F. Farrell, H.W. Anderson, JR., P.E. Felsheim. 1975. Water resources of the Root River Watershed, southeastern Minnesota. Hydrologic Investigations Atlas HA-548, U.S. Geological Survey, Reston, VA.

Clifton, C. 1989. Effects of vegetation and land use on channel morphology. Pages 121-129, in R.E. Gresswell, B.A. Barton, J.L. Kershner (editors), Practical approaches to riparian resource management, An Educational Workshop, May 8-11, U.S. Bureau of Land Management, Billings, MT.

Conover, W.J. 1980. Practical non-parametric statistics. 2nd Edition, John Wiley and Sons, New York.

Corkum, L.D. 1989. Patterns of benthic invertebrate assemblages in rivers of northwestern North America. Freshwater Biology 21: 191-205.

Corkum, L.D. and J.J.H. Ciborowski. 1988. Use of alternative classifications in studying broad-scale distributional patterns of lotic invertebrates. Journal of the North American Benthological Society 7: 167-179.

Cummins, K.W. 1988. The study of stream ecosystems: a functional view. Pages 247-258, in L.R. Pomeroy and

J.J. Alberts (editors). Concepts of ecosystem ecology. Springer-Verlag, New York.

Cummins, K.W., M.A. Wilzbach, D.M. Gates, J.B. Perry, W.B. Taliaferro. 1989. Shredders and riparian vegetation. Bioscience 39: 24-30.

Dance, K.W. and H.B.N. Hynes. 1980. Some effects of agricultural land use on stream insect communities. Environmental Pollution (Series A) 22: 19-28.

Delcourt, P.A. and H.R. Delcourt. 1988. Quaternary landscape ecology: Relevant scales in space and time. Landscape Ecology 2: 23-44.

Fisher, S.G. 1987. Succession, scale and hypothesis testing in streams. Canadian Journal of Fisheries and Aquatic Science 44: 689.

Frissell, C.A., W.J. Liss, C.E. Warren, M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10: 199-214.

Gallant, A.L., T.R. Whittier, D.P. Larsen, J.M. Omernik, R.M. Hughes. 1989. Regionalization as a tool for managing environmental resources. U.S. EPA 600/3-89/060, Environmental Research Laboratory, Corvallis, OR.

Green, D.M. and J.B. Kauffman. 1989. Nutrient cycling at the land-water interface: The importance of the riparian zone. Pages 61-68, in R.E. Gresswell, B.A. Barton, J.L. Kershner (editors), Practical approaches to riparian resource management, An Educational Workshop, May 8-11, U.S. Bureau of Land Management, Billings, MT.

Hallberg, G.R., R.D. Libra, B.E. Hoyer. 1985. Nonpoint source contamination of ground water in karst-carbonate aquifers in Iowa. Pages 109-115, in Perspectives on nonpoint source pollution, Proceedings of National Conference, National Water Well Association, October 28-30, Bowling Green, KY.

Hawkes, C.L., D.L. Miller, W.G. Layher. 1986. Fish ecoregions of Kansas: stream fish assemblage patterns and associated environmental correlates. Environmental Biology of Fishes 17: 267-279.

Heiskary, S.A., C.B. Wilson, D.P. Larsen. 1987. Analysis of regional patterns in lake water quality: Using ecoregions for lake management in Minnesota. Lake and Reservoir Management 3: 337-344.

Hilsenhoff, W.L. 1982. Using a biotic index to evaluate water quality in streams. Wisconsin Department of Natural Resources, Technical Bulletin Number 132, Madison, WI.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20: 31-39.

Hite, R.L. and B.A. Bertrand. 1989. Biological stream characterization (BSC): A biological assessment of Illinois stream quality. Special Report Number 13 of the Illinois State Water Plan Task Force, Illinois Environmental Protection Agency, Springfield, IL.

Hughes, R.M. and J.M. Omernik. 1981. A proposed approach to determine regional patterns in aquatic ecosystems. Pages 92-102, in N.B. Armentrout (editor). Acquisition and Utilization of Aquatic Habitat Inventory Information. Proceedings of a Symposium, American Fisheries Society, Bathesda.

Hughes, R.M., E. Rexstad, C.E. Bond. 1987. The relationship of aquatic ecoregions, river basins and physiographic provinces to the ichthyogeographic regions of Oregon. Copeia 1987: 423-432.

Hynes, H.B.N. 1975. The stream and its valley. International Association of Theoretical and Applied Limnology 19: 1-15.

Jeffers, J.N.R. 1988. Statistical and mathematical approaches to issues of scales in ecology. Pages 47-56, in T. Rosswall, R.G. Woodmansee, P.G. Risser (editors). Scales and global change. John Wiley and Sons, New York.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management 5: 55-68.

Karr, J.R. and I.J. Schlosser. 1978. Water resources and the land-water interface. Science 201: 229-234.

Koestler, A. 1967. The ghost in the machine. MacMillan Company, New York.

Kolasa, J. 1989. Ecological systems in hierarchical perspective: Breaks in community structure and other consequences. Ecology 70: 36-47.

Kratz, T.K. and G.L. Jensen. 1977. An ecological geographic division of Minnesota. Minnesota Department of Natural Resources, St. Paul, MN.

Kuehnast, E.L. 1974. The climate of Minnesota. Pages 706-742, in Officials of the U.S. National Oceanic and Atmospheric Administration, Climates of the United States (Volume 2). U.S. Department of Commerce.

Larsen, D.P., D.R. Dudley, R.M. Hughes. 1988. A regional approach to assess attainable water quality: An Ohio case study. Journal of Soil and Water Conservation 43: 171-176.

LeGrand, H.E. 1973. Hydrological and ecological problems of karst regions. Science 179: 859-864.

Lenat, D.R. 1984. Agriculture and stream water quality: a biological evaluation of erosion control practices. Environmental Management 8: 333-344.

Lenat, D.R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7: 222-233.

Lenat, D.R., L.A. Smock, D.L. Penrose. 1980. Use of benthic macroinvertebrates as indicators of environmental quality. Pages 97-112, in D.L. Worf (editor). Biological monitoring for environmental effects. Lexington Books, D.C. Heath and Company, Lexington.

Lyons, J. 1989. Correspondence between the distribution of fish assemblages in Wisconsin streams and Omernik's ecoregions. American Midland Naturalist 122: 163-182.

May, R.M. 1989. Levels of organization in ecology. Pages 339-363, in J.M. Cherrett (editor). Ecological concepts. Blackwell Scientific Publications, Oxford.

Menzel, B.W., J.B. Barnum, L.M. Antosch. 1984. Ecological alterations of Iowa prairie-agricultural streams. Iowa State Journal of Research 59: 5-30.

Minnesota Pollution Control Agency. 1990. Standards for the protection of the quality and purity of the waters of the state. Minnesota Rules Chapter 7050, Minnesota Pollution Control Agency, St. Paul, MN.

Minnesota State Planning Agency. 1971. State of Minnesota land-use. Minnesota Land Management Information System, Minnesota State Planning Agency. (map)

Minshall, G.W. 1988. Stream ecosystem theory: a global perspective. Journal of the North American Benthological Society 7: 263-288.

NH Analytical Software. 1988. Statistix II: An interactive statistical analysis program for microcomputers. NH Analytical Software, Roseville, MN.

Norusis, M.J. 1988. SPSS/PC+ advanced statistics V2.0 for the IBM PC/XT/AT and PS/2. SPSS Inc., Chicago, IL.

Ojakangas, R.W. and C.L. Matsch. 1982. Minnesota's geology. University of Minnesota Press, Minneapolis, MN.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. The Annals of the Association of American Geographers 77: 118-125.

Omernik, J.M. and A.L. Gallant. 1988. Ecoregions of the upper midwest states. EPA 600/3-88/037, Environmental Research Laboratory, Corvallis, OR.

Omernik, J.M. and G.E. Griffith. 1986. Total alkalinity of surface waters: a map of the upper midwest region of the United States. Environmental Management 10: 829-839.

O'Neill, R.V. 1988. Hierarchy theory and global change. Pages 29-45, in T. Rosswall, R.G. Woodmansee, P.G. Risser (editors). Scales and global change. John Wiley and Sons, New York.

O'Neill, R.V., D.L. DeAngelis, J.B. Waide. 1986. A hierarchical concept of ecosystems. Princeton University Press, Princeton.

Peckarsky, B.L. 1986. Colonization of natural substrates by stream benthos. Canadian Journal of Fisheries and Aquatic Science 43: 700-709.

Peckarsky, B.L. 1987. Succession, scale, and hypothesis testing in streams: A reply to Fisher. Canadian Journal of Fisheries and Aquatic Science 44: 689-691.

Perry, J.A., R.C. Ward, J.C. Loftis. 1984. Survey of state water quality monitoring programs. Environmental Management 8: 21-26.

Perry, J.A., D.J. Schaeffer, H.K. Kerster, E.E. Herricks. 1985. The Environmental Audit II: Application to stream network design. Environmental Management 9: 199-208.

Perry, J.A., N.H. Troelstrup, Jr., W. Bartodziej, T.F. Wilton. 1988. Risks to surface water quality in the Lanesboro Watershed: Southeastern Minnesota. Department of Forest Resources Staff Paper Series Number 63, College of Forestry, University of Minnesota, St. Paul, MN.

Peterjohn, W.T. and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 65: 1466-1475.

Pinay, G. and H. Decamps. 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: A conceptual model. Regulated Rivers: Research and Management 2: 507-516.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: Benthic macroinvertebrates and fish. EPA /444/4-89-001, U.S. Environmental Protection Agency, Assessment and Watershed Protection Division, Washington, D.C.

Resh, V.H. and D.M. Rosenberg. 1989. Spatial-temporal variability and the study of aquatic insects. Canadian Entomologist 121: 941-963.

Rohm, C.M., J.W. Giese, C.C. Bennett. 1987. Evaluation of an aquatic ecoregion classification of streams in Arkansas. Journal of Freshwater Ecology 4: 127-140.

Ross, H.H. 1963. Stream communities and terrestrial biomes. Archiv fur Hydrobiologie 59: 235-242.

Schaeffer, D.J., H.K. Kerster, J.A. Perry, S.K. Sokolik, D.K. Cox. 1985. The Environmental Audit I: Concepts. Environmental Management 9: 191-198.

Schumm, S.A. 1988. Variability of the fluvial system in space and time. Pages 225-250, in T. Rosswall, R.G. Woodmansee, P.G. Risser (editors). Scales and global change. John Wiley and Sons, New York.

Singer, R.D., M.T. Osterholm, C.P. Straub. 1982. Groundwater quality in southeastern Minnesota. Bulletin Number 109, Water Resources REsearch Center, University of Minnesota, St. Paul, MN.

Smart, M.M., J.R. Jones, J.L. Sebaugh. 1985. Stream-watershed relations in the Missouri Ozark plateau province. Journal of Environmental Quality 14: 77-82.

St. Ores, J.S., E.C. Alexander, Jr., C.F. Halsey. 1982. Groundwater pollution prevention in southeast Minnesota's karst region. Bulletin Number 465, Agricultural Extension Service, University of Minnesota, St. Paul, MN.

Swanson, F.J., S.V. Gregory, J.R. Sedell, A.G. Campbell. 1982. Land-water interactions: The riparian zone. Pages 267-291, in R.L. Camonas (editor). Analysis of coniferous forest ecosystems in the western United States. Hutchinson Ross Publishing Co.

Townsend, C.R. 1989. The patch dynamics concept of stream community ecology. Journal of the North American Benthological Society 8: 36-50.

Troelstrup, N.H., Jr. and J.A. Perry. 1989. Water quality in southeastern Minnesota streams: Observations along a gradient of land use and geology. Journal of the Minnesota Academy of Science 55: 6-13.

University of Minnesota. 1973. Minnesota soil atlas. St. Paul Sheet. Miscellaneous Report 120, Agricultural Experiment Station, University of Minnesota, St. Paul, MN. (map).

United States Department of Agriculture and Minnesota Department of Agriculture. 1988. Minnesota agriculture statistics 1988. Minnesota Agricultural Statistics Service, St. Paul, MN.

Vitousek, P.M. and J.M. Melillo. 1979. Nitrate losses from disturbed forests: Patterns and mechanisms. Forest Science 25: 605-619.

Wall, D.B., S.A. McGuire, J.A. Magner. 1989. Water quality monitoring and assessment in the Garvin Brook Clean Water Project Area. Minnesota Pollution Control Agency, St. Paul, MN.

Ward, J.V. 1989. The four-dimensional nature of lotic ecosystems. Journal of the North American Benthological Society 8: 2-8.

Weisberg, S. 1985. Applied linear regression. 2nd edition, John Wiley & Sons, New York.

Welch, H.E., P.E.K. Symons, D.W. Narver. 1977. Some effects of potato farming and forest clearcutting on small New Brunswick streams. Fisheries and Marine Service Technical Report 745.

Whittier, T.R., R.M. Hughes, D.P. Larsen. 1988. Correspondence between ecoregions and spatial patterns in stream ecosystems in Oregon. Canadian Journal of Fisheries and Aquatic Science 45: 1264-1278.

Wiens, J.A. 1989. Spatial scaling in ecology. Functional Ecology 3: 385-397.

Wilton, T.F. and J.A. Perry. 1989. Litter decomposition dynamics in karst stream ecosystems. Unpublished Plan B Master's Paper, Department of Forest

Resources, University of Minnesota, St. Paul, MN.

Winchell, N.H. and W. Upham. 1884. Geology of Minnesota. Volume I of the final report. Johnson, Smith and Harrison, State Printers, Minneapolis, MN.

Woodall, W.R., Jr. and J.B. Wallace. 1972. The benthic fauna in four small southern Appalachian streams. American Midland Naturalist 88: 393-407.